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Health and Environmental Effects Profile for Fluridone

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HEALTH AND ENVIRONMENTAL EFFECTS PROFILE FOR FLURIDONE

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16. ABSTRACT

The Health and Environmental Effects Profile for fluridone was prepared by the Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, OH for the Office of Solid Waste to support listings of hazardous constituents of a wide range of waste streams under Section 3001 of the Resource Conservation and Recovery Act (RCRA). Both published literature and information obtained from Agency program office files were evaluated as they pertained to potential human health, aquatic life and environmental effects of hazardous waste constituents.

Quantitative estimates have been presented provided sufficient data are available. Fluridone has been determined to be a systemic toxicant. An Acceptable Daily Intake (ADI), defined as the amount of a chemical to which humans can be exposed on a daily basis over an extended period of time (usually a lifetime) without suffering a deleterious effect, for fluridone is 9×10^{-2} (mg/kg bw/day) for oral exposure.

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PREFACE

Health and Environmental Effects Profiles (HEEPs) are prepared for the Office of Solid Waste by the Office of Health and Environmental Assessment. The HEEPs are intended to support listings of hazardous constituents of a wide range of waste streams under Section 3001 of the Resource Conservation and Recovery Act (RCRA). Both published literature and information obtained from Agency program office files are evaluated as they pertain to potential human health, aquatic life and environmental effects of hazardous waste constituents. The literature searched and the dates of the searches are included in the section titled "Appendix: Literature Searched."

Quantitative estimates are presented provided sufficient data are available. For systemic toxicants, these include, Acceptable Daily Intakes (ADIs) for chronic exposures. An ADI is defined as the amount of a chemical to which humans can be exposed on a daily basis over an extended period of time (usually a lifetime) without suffering a deleterious effect. In the case of suspected carcinogens, ADIs are not estimated in this document series. Instead, a carcinogenic potency factor or q_1^* is provided. These potency estimates are derived for both oral and inhalation exposures where possible. In addition, unit risk estimates for air and drinking water are presented based on inhalation and oral data, respectively.

The first draft of this document was prepared by Syracuse Research Corporation under EPA Contract No. 68-03-3228. The document was subsequently revised after reviews by staff within the Office of Health and Environmental Assessment: Carcinogen Assessment Group, Reproductive Effects Assessment Group, Exposure Assessment Group, and the Environmental Criteria and Assessment Office in Cincinnati.

The HEEPs will become part of the EPA RCRA docket.

ACKNOWLEDGEMENTS

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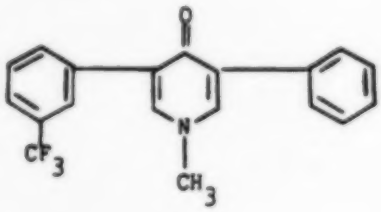
LIST OF ABBREVIATIONS

BCF	Bioconcentration factor
bw	Body weight
LD_{50}	Dose lethal to 50% of the recipients (and all other subscripted dose levels)
ppm	Parts per million
UV	Ultraviolet

1. INTRODUCTION

1.1. STRUCTURE AND CAS NUMBER

Fluridone is the common name for 1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4(1H)-pyridinone. It was introduced in 1976 by Eli Lilly and Company under the code number "EL-171" (Waldrep and Taylor, 1976). Fluridone is the active ingredient in the aquatic herbicide "SONAR" (West et al., 1983). The structure and molecular formula for fluridone are as follows:



Empirical formula: $C_{19}H_{14}F_3NO$

Molecular weight: 329.3

The Chemical Abstract Services (CAS) Registry number for fluridone is 59756-60-4.

1.2. PHYSICAL AND CHEMICAL PROPERTIES

Fluridone is a white crystalline solid at ordinary temperatures (WSSA, 1979). Selected physical properties are listed below:

Melting point:	154-155°C	(WSSA, 1979)
Vapor pressure (25°C):	$<1 \times 10^{-7}$ mm Hg	(WSSA, 1979)
Solubilities (25°C):		(WSSA, 1979)
methanol	>1 g/100 ml	
diethyl ether	>0.1 g/100 ml	
ethyl acetate	>0.5 g/100 ml	
chloroform	>1 g/100 ml	
hexane	<500 ppm	
water	12 ppm	

Log octanol/water partition coefficient:	1.87	(West et al., 1983)
Average BCF for fish in natural environment:	1.33-6.08	(West et al., 1983)
pK_A value:	1.7	(Weber, 1980)

Fluridone is moderately soluble in chloroform, ethanol, acetone and ethylacetate, somewhat less soluble in ether and benzene, and almost insoluble in hexane (Waldrep and Taylor, 1976). It has a half-life of 23 hours in deionized water exposed to UV irradiation (WSSA, 1979).

1.3. PRODUCTION DATA

Fluridone is synthesized by reacting benzyl(3-trifluoromethyl)benzyl ketone with ethyl formic acid and methylamine hydrochloride (Taylor, 1976, 1977).

The only domestic producer of fluridone is Eli Lilly and Co. in Lafayette, IN (SRI, 1983). Current production figures are not available. USITC (1983) does not have a listing for fluridone, which may indicate that production and sales of fluridone in 1982 did not exceed 5000 pounds or *5000. West et al. (1983) has noted that the aquatic herbicide fluridone is still experimental.

1.4. USE DATA

Fluridone is a herbicide that controls most annual grasses and many broadleaf weeds (WSSA, 1979). Cotton is the only annual crop to date which demonstrates physiological tolerance to fluridone. It can be incorporated into the soil or applied to the soil surface as a spray. Recommended rates of application range from 0.2-1 lb/acre, with water as the usual carrier.

Due to soil residue problems, the major use of fluridone in the future is likely to be as an aquatic herbicide (Shea and Weber, 1983a). Fluridone has been found to control vascular aquatic weeds at low application rates (West et al., 1983).

2. ENVIRONMENTAL FATE AND TRANSPORT PROCESSES

2.1. WATER

The fate of fluridone in the aquatic environment will depend on its ability to undergo photochemical, chemical and biological reactions, and partition to the sediments and atmosphere. The photodegradation of fluridone in water has been studied by several investigators. West et al. (1979) exposed an aqueous solution of fluridone in deionized water to four sunlamps and four black lights in a light box experiment. The fluridone exhibited a photodecomposition half-life of ~23 hours. Addition of a 2% acetone photosensitizer decreased the half-life to ~6 hours. Muir and Grift (1982) exposed pond water, containing 5 ppm fluridone, to sunlight for 8 days in stoppered pyrex flasks. The half-life of the fluridone in the flasks was found to be ~6 days. In control flasks covered with aluminum foil to prevent irradiation, fluridone degradation was negligible (<5%) over the 8-day period. The major identifiable photolysis products in the flasks were found to be 2-hydroxy- and 4-hydroxyfluridone, dephenylfluridone and fluridone-acid. Saunders and Mosier (1983) conducted laboratory photolysis studies of fluridone under a variety of conditions using sunlamps and black lights as the irradiating source. The measured half-lives of fluridone under these conditions are summarized below:

Solution Type	Half-life in Hours
water, 1.0 ppm concentration	34
water, 0.2 ppm concentration	26
water, nitrogen purged	22
pH 3, buffer	30
pH 6, buffer	55
ph 9, buffer	28

The above results suggest that the half-life of fluridone depends only slightly on the concentration and pH of the solution. Saunders and Mosier

(1983) also conducted photolysis tests on distilled vs. lake water and found virtually no difference in the photolysis rates. The photolysis products of fluridone decomposition were analyzed by Saunders and Mosier (1983); a schematic representation is diagrammed in Figure 2-1. Compounds I and II were detected in only small amounts; therefore, it was suggested that fluridone photolysis proceeded by a mechanism that degraded the pyridinone ring. West et al. (1983) noted that the photo products appear to dissipate rapidly from water via volatilization so that significant levels do not accumulate in the water.

Fluridone does not hydrolyze in aqueous solution (West et al., 1983).

Muir and Grift (1982) examined the microbial degradation of fluridone in water-sediment culture flasks over a 26-month period. Using three different sediment types, an average half-life of 12 months was determined. After 26 months <13% of the ^{14}C in the sediment was fluridone, while in autoclaved sediments, in the control, almost all of the ^{14}C was fluridnne. This suggests that fluridone can be degraded by microorganisms; however, the microbial degradation rate is not nearly as rapid as the photolysis rate, but probably is important to the degradation of fluridone in sediments where photolysis cannot take place. In a similar experiment, Marquis et al. (1982) obtained a sandy loam and a silt loam soil from the bottom of an irrigation canal, treated them with 10 ppm fluridone, placed the treated soils in laboratory beakers and covered the soils with tap water. The beakers were totally covered with foil to exclude light and incubated at room temperature for periods up to 12 months. After 12 months, ~30% of the applied fluridone remained in the soils. The authors suggested that microbial metabolism was responsible for the fluridone degradation; an autoclaved control test was not performed.

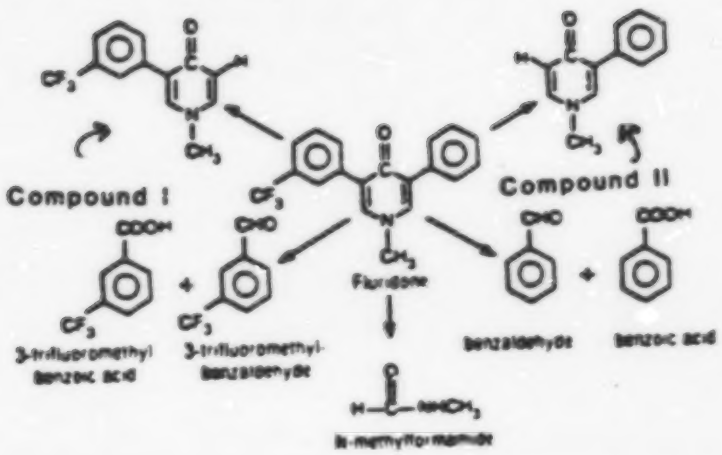


FIGURE 2-1

Photolysis Products of Fluridone

(Saunders and Mosier, 1983)

The loss of fluridone from water due to volatilization has been estimated by Muir and Grift (1982). Henry's constant was calculated from the rate of loss of fluridone from a column of distilled water sparged with nitrogen. The half-life of fluridone in a pond (0.5 m depth; 10-m^2 surface area) was then calculated to be 50.8 days. The actual rate of loss would be expected to be reduced by the portion of fluridone sorbed to bottom sediments.

Fluridone is sorbed onto sediment present in the aquatic environment, and deposition of fluridone in the hydrosoil has been demonstrated (West et al., 1979, 1983; West and Parka, 1981; Muir et al., 1980; Muir and Grift, 1982). The range of fluridone present in the hydrosoil, as a percentage of total fluridone applied to the aquatic ecosystem, was ~1-27% and varied with time and conditions.

The bioaccumulation of fluridone has been examined by several investigators. West et al. (1983) studied the bioconcentration of fluridone and its major metabolites in fish in a total of 40 pond and lake trials. The average BCF for fluridone plus its major metabolite (4-hydroxyfluridone) in 10 fish species collected during the field trials (175 samples) were 1.33, 7.38 and 6.08 in edible, inedible and whole fish tissues, respectively. BCF is defined as the ppm concentration of the residue in fish divided by the ppm concentration in water. Residues in fish collected from lakes were lower than residues in fish collected from ponds. This circumstance corresponded to a lower fluridone concentration in the lake water. West et al. (1983) also noted that the log octanol-water partition coefficient for fluridone (1.87) predicts a BCF value in whole fish of 7.8. This value is consistent with their experimental results and indicates that fluridone will

not bioconcentrate to a significant degree in the aquatic food chain. Muir et al. (1982) measured the uptake and bioconcentration of fluridone in rainbow trout placed in a glass aquarium. A BCF value of 91 was determined, suggesting a low bioconcentration potential. The fish were then placed in clean flowing water and tested for clearance; 80-90% of the fluridone plus metabolites was eliminated within 4 days. Muir et al. (1980) applied fluridone to three small ponds at varying concentrations and measured the fluridone levels in minnows over a 70-week period. During the sampling period, fluridone levels in the minnows were very low (<0.02-0.14 µg/g) with BCF values ranging from 0-64. Similar results, demonstrating the low potential for fluridone bioconcentration in fish, were obtained by West et al. (1979) in field experiments conducted in Michigan, New York, Florida and the Panama Canal.

West et al. (1979) applied fluridone to ponds in Michigan, New York, Florida and the Panama Canal and found rapid dissipation with a half-life averaging 5 days. These findings suggest that dissipation was the result of photodecomposition, sedimentation and uptake by aquatic plants. Dissipation of fluridone from hydrosoils is variable and slow compared to its dissipation from water. Muir et al. (1980) found the persistence of flurodine, applied to three small experimental ponds, to have a water half-life of 4-7 days; however, the persistence of fluridone in the hydrosoils was found to be much longer, with a half-life of 1 year or more at varying treatment concentrations.

West and Parka (1981) measured the rate of disappearance of fluridone from water and hydrosoil in two ponds using both surface application and bottom application. No significant differences were reported since the water half-life was 21 days for surface application and 26 days for bottom

application. No detectable fluridone was found in the hydrosoils 56 days after treatment. Only 4% of the initial fluridone concentration remained in the pond water 110 days after treatments.

Muir and Grift (1982) studied the persistence of fluridone both in the laboratory, in culture flasks, and in the field, in small ponds. The half-life of fluridone in sediment was found to be ~1 year under laboratory conditions (25°C) and ~0.33 years under field conditions. The half-life of fluridone in ponds treated with 0.1 ppm fluridone was ~2-3.5 days. West et al. (1983) examined fluridone dissipation patterns in 40 pond and lake experiments in the United States, Canada and Panama. Fluridone was found to have an average half-life of 20 days in pond water and 3 months in pond hydrosoil. In lake trials, half-lives could not be estimated accurately because dissipation was due largely to dispersal and dilution rather than degradation or sedimentation.

2.2. AIR

Data specific to the fate and transport of fluridone in the air were not located in the available literature cited in the Appendix. Data presented in Section 2.1. indicate that fluridone is photolabile in aqueous solution to a significant extent, which may suggest that photodegradation in the atmosphere has potential importance. Cupitt (1980) suggested that adsorption will be a reasonable vapor-phase removal mechanism only for materials with saturation vapor pressures of $<10^{-7}$ mm Hg. Since fluridone has a saturation vapor pressure $<10^{-7}$ mm Hg at 25°C (WSSA, 1979), this compound may become, and remain, sorbed to particulate matter in the air.

2.3. SOIL

Microbial degradation of fluridone via soil microorganisms has been demonstrated. In a laboratory growth chamber experiment, Banks et al.

(1979) treated four different soils with 1 ppm fluridone and measured the fluridone degradation for a period of 270 days. Replications of the same soils were sterilized before fluridone treatment as a control to examine microbial activity. In two of the soils (Brennan fine sandy loam and Hidalgo sandy clay loam), degradation was significantly faster in the nonsterile soil compared to the sterile soil. For example, after 60 days in the Brennan fine sandy loam, 65% of the initial fluridone remained in the sterilized sample while only 25% of the initial fluridone remained in the nonsterilized sample. In the other two soils tested (Miller clay and Lufkin fine sandy loam), however, degradation of fluridone was virtually the same in sterile vs. nonsterile samples. The authors suggested that the data indicated that microbial breakdown of fluridone was important but may not predominate in all soils. As noted in Section 2.1., studies by Muir and Grift (1982) and Marquis et al. (1982) have demonstrated the microbial degradation of fluridone.

The photolability of fluridone under aqueous conditions is environmentally significant (see Section 2.1.). Therefore, it is possible that photodegradation of fluridone from soil surfaces contributes to its overall loss from the soil, although the importance of this mechanism has not been investigated in the available literature cited in the Appendix.

The chemical hydrolysis of fluridone in soil has not been discussed in the available literature cited in the Appendix.

The adsorption and leaching of fluridone in soil has been examined by several authors. Banks (1979) and Banks and Merkle (1979) performed leaching tests with fluridone in laboratory soil columns on three soil types. The fluridone moved 12-16 cm deep in coarse loamy sand but moved only 1-2 cm deep in fine sandy loam or clay soils. The movement appeared to be related

to the organic matter and clay content; movement decreased as organic matter content increased. Weber (1980) found that fluridone is readily adsorbed by organic matter in soil and by Ca-montmorillonite (to a lesser extent), but poorly adsorbed, by a relative measure, to sandy loam soil. It was also determined that adsorption of fluridone by soil matter increased with decreasing pH. Similar results were reported by Shea (1981), Shea and Weber (1980), and Shea and Weber (1983a). Baumann (1981) studied the leaching of fluridone in five Texas soils in soil column studies and in field studies. R_f values were calculated for fluridone movement via soil thin-layer plate chromatography; calculated R_f values were 0.66, 0.22, 0.18, 0.15 and 0.06 for Brownfield sand soil, Amarillo fine sandy loam I, Amarillo fine sandy loam II, Olton loam and Randall clay, respectively. Field tests confirmed the mobility pattern resulting from the R_f values calculated. Increasing the water volume in the leaching columns enhanced the fluridone mobility. Loh et al. (1979) found a correlation between the organic matter content and soil adsorption coefficient of 14 different soils and the phytotoxicity of applied fluridone. Phytotoxicity decreased as fluridone adsorption increased or as organic matter content increased.

WSSA (1979) reported that the persistence of fluridone is complex. In most cotton producing areas, residues may carry over to the next cropping season and cause slight injury to other crops planted in rotation. Banks (1979) and Banks et al. (1979) measured the persistence of fluridone under field conditions and found that 25% of the applied fluridone remained 385 days after application in fine sandy loam, while only 5% of applied fluridone remained in clay soil after 250 days. Fluridone degradation was additionally found to be slower in the same soils under controlled laboratory conditions than in the field. Also, soil incorporation increased the

persistence in the sandy loam soil but not in the clay soil. Shea and Weber (1983b) examined the persistence of fluridone during greenhouse studies by chlorophyll measurements. Using a Norfolk sandy loam soil, a half-life of 18 days was calculated for the upper 15 cm of soil depth in the absence of cultivation. Subsequent cultivation 58 days after treatment suggested that significant levels of fluridone were still present in the soil; fluridone may have been leached below the 15 cm depth examined. Shea and Weber (1980) found little residual fluridone activity in greenhouse bioassays 20.5-13 weeks after application to various soils. Aharonson et al. (1983) found that fluridone disappeared slower in solar-heated soils than in non-heated soils.

3. EXPOSURE

3.1. WATER

Fluridone is not listed in the U.S. EPA STORET Data Base. Monitoring data, other than that generated from intentional application (see Section 2.1.), for fluridone exposure in water were not located in the available literature cited in the Appendix.

3.2. FOOD

Pertinent data regarding exposure to fluridone in food were not located in the available literature cited in the Appendix.

3.3. INHALATION

Pertinent data regarding inhalation exposure to fluridone were not located in the available literature cited in the Appendix.

3.4. DERMAL

Pertinent data regarding dermal exposure to fluridone were not located in the available literature cited in the Appendix.

Fluridone is an experimental herbicide at present (West et al., 1983) with no projected use on cultivated food crops. Also, data presented in Section 2.1. indicates that fluridone is not bioconcentrated to significant quantities in fish.

4. PHARMACOKINETICS

Pertinent data regarding the pharmacokinetics of fluridone were not located in the available literature cited in the Appendix.

5. EFFECTS

5.1. CARCINOGENICITY

Pertinent data regarding the carcinogenicity of fluridone were not located in the available literature cited in the Appendix.

5.2. MUTAGENICITY

Pertient data regarding the mutagenicity of fluridone were not located in the available literature cited in the Appendix.

5.3. TERATOGENICITY

Pertinent data regarding the teratogenicity of fluridone were not located in the available literature cited in the Appendix.

5.4. OTHER REPRODUCTIVE EFFECTS

Pertinent data regarding other reproductive effects of fluridone were not located in the available literature cited in the Appendix.

5.5. CHRONIC AND SUBCHRONIC TOXICITY

Rats and mice exposed orally to diets containing 2000 ppm fluridone for 90 days displayed "slight histological changes in livers and kidneys" but no other signs of toxicity. Dogs exposed orally to either 50, 100 or 200 mg/kg fluridone for 90 days exhibited no adverse effects to treatment. This information was provided by Elanco, the manufacturer (WSSA, 1979). No other information was located in the available literature cited in the Appendix.

5.6. OTHER RELEVANT INFORMATION

Data regarding the acute toxicity of fluridone are summarized in Table 5-1. No cases of human poisoning have been reported or observed (WSSA, 1979).

TABLE 5-1

Acute Toxicity of Fluridone*

Species	Variable	Value (mg/kg)
Rat	LD_0	>10 g/kg
Mouse	LD_{50}	>10 g/kg
Dog	LD_0	>500 mg/kg
Cat	LD_0	>250 mg/kg

*Source: WSSA, 1979

6. AQUATIC

6.1. ACUTE

Although laboratory studies concerning acute toxicity of fluridone were not located in the literature searched, the results of several field investigations indicate that single treatments of ponds with this aquatic herbicide had little, if any, effect on aquatic animals.

Grant et al. (1979) treated ponds with 0.75-2 lb/acre and stated that fluridone did not adversely affect wildlife in the ponds. Sanders et al. (1979) reported that there were no noticeable disturbances of the plankton and benthic animal populations in ponds treated with as much as 6.72 kg/ha of two fluridone formulations. Arnold (1979) stated that zooplankton and benthic populations changed very little after treatment of ponds with 1.12 kg/ha. Rivera et al. (1979) reported that treatment with 0.5 lb/acre had no effect on nontarget aquatic life. As discussed in Section 6.3., all of these pond treatments effectively controlled aquatic weeds. The results of these studies indicate that concentrations that control aquatic plants have little or no effect on aquatic animals.

6.2. CHRONIC

Pertinent data regarding the chronic toxicity of fluridone to aquatic animals were not located in the available literature cited in the Appendix. As discussed previously, single fluridone treatments in ponds did not have long-term effects on aquatic animals.

6.3. PLANT EFFECTS

Numerous studies concerning the effects of fluridone on aquatic plants are available because this chemical has been evaluated extensively as an aquatic herbicide. Fluridone inhibits carotenoid synthesis causing chlorosis (bleaching) in new growth of aquatic macrophyte plants (McCowen et al.,

1979). Exposure of sago pondweed to 5 and 10 ppm for 24 hours resulted in 50 and 75% mortality, respectively (Irigoyen and Brevedan, 1983). Anderson (1981) reported that 10 days of exposure to 1 ppm caused a 50 and 87% decrease in growth of sago pondweed and American pondweed, respectively. Taylor (1980) found that 0.3 ppm fluridone totally controlled four species of macrophytes in an artificial pond. Treatments of test plots with fluridone at several concentrations resulted in reduction of Hydrilla biomass after 84 days at doses of >1.70 kg/ha, but not at 0.84 kg/ha (Sanders et al., 1979). Complete control of several aquatic macrophyte species was obtained in field trials at treatment concentrations of 1-4 ppm (McCowen et al., 1979), 1.12 kg/ha (Arnold, 1979), 0.5 lb/acre (Rivera et al., 1979) and 0.75-2 lb/acre (Grant et al., 1979).

6.4. RESIDUES

Several studies have been conducted to determine the residue dynamics of fluridone in aquatic animals after treatment of ponds for aquatic plant control. Muir et al. (1980) treated small ponds with 0.07-5 ppm fluridone and reported that residues in fathead minnows were between <0.02-0.14 ppm over a 70-week period, with BCFs ranging from 0-64. Grant et al. (1979) reported that treatment of ponds with 0.75-2 lb/acre resulted in maximum residues in fish 3-14 days after treatment. After 120 days, fluridone was detectable in channel catfish (0.015 ppm) but not in largemouth bass or bluegill sunfish.

As a part of 40 lake and pond experiments, West et al. (1983) investigated the accumulation of fluridone and its major metabolite (4-hydroxyfluridone) in 10 fish species. Maximum residues in whole fish (0.4 ppm) occurred between 2 and 4 weeks after treatment with 0.84-2.24 kg/ha. Maximum BCFs ranged between 1.59 and 15.51 for the fish species and averaged

6.08 for all species. The concentrations of fluridone and 4-hydroxyfluridone in the fish were approximately equal, indicating that substantial metabolism had occurred. The average total residue concentration in fish was <0.01 ppm 4 months after treatment.

The only available laboratory investigation of fluridone bioconcentration was that of Muir et al. (1982) who measured the uptake and elimination of ^{14}C-carbonyl-labeled fluridone in rainbow trout, Salmo gairdneri, and midge larvae, Chironomus tentans. The equilibrium BCFs, calculated from uptake and elimination rate constants, were 91 for trout and 128 for midge larvae. Elimination after transfer to clean flowing water was rapid, with 80-90% of the radioactivity being eliminated within 4 days by trout and within 4 hours by midge larvae. The BCFs reported in this study are considerably higher than those obtained in the previously mentioned field studies, possibly because they were calculated on the basis of total ^{14}C content rather than the concentration of unmetabolized parent compound.

The residues from these residue studies indicate that fluridone is rapidly metabolized and eliminated by aquatic animals and has a relatively low bioconcentration potential.

6.5. OTHER RELEVANT INFORMATION

Other relevant information regarding the effects and residues of fluridone in aquatic organisms was not located in the available literature cited in the Appendix.

7. EXISTING GUIDELINES AND STANDARDS

7.1. HUMAN

The U.S. EPA has established a temporary tolerance for fluridone of 0.01 ppm in potable water (Code of Federal Regulations, 1981). These levels are tolerated only in conjunction with the experimental application of fluridone to lakes and ponds by the Elanco Products Company, when applied legally under the provisions of an experimental use permit. This permit expired on June 1, 1983. A petition has also been granted for a temporary tolerance of 0.05 ppm of fluridone on cottonseed, in order to allow the marketing of this agricultural product following treatment with fluridone in accordance with the experimental use permit (Federal Register, 1979). This permit expired on July 1, 1980.

7.2. AQUATIC

Existing guidelines and standards for the protection of nontarget aquatic organisms from the effects of fluridone were not located in the available literature.

8. RISK ASSESSMENT

No data were located in the available literature regarding the carcinogenicity, mutagenicity, teratogenicity or reproductive effects of fluridone. The only systemic toxicity data located consisted of unpublished subchronic (90-day) toxicity studies in rats, mice and dogs (WSSA, 1979). In rats and mice, histologic changes in the liver and kidney were reported after exposure to diets containing 2000 ppm of fluridone. Effects at other treatment levels were not reported. In dogs, no effects were observed at the highest dose tested, 200 mg/kg bw/day. The lack of detail available in the summaries (WSSA, 1979) of these unpublished studies precludes the use of these data in risk assessment.

Statements concerning available literature in this document refer to published, quotable sources and are in no way meant to imply that confidential data, which this document could not address, are not in existence. These data, which are considered Confidential Business Information (CBI), are available in summary form as a CBI Appendix that can be accessed by individuals with appropriate clearance. Conclusions from the CBI Appendix regarding risk assessment can, however, be presented in this chapter.

The CBI Appendix to this document reviews in more detail the subchronic studies in mice, rats and dogs mentioned above. The CBI Appendix also reviews other animal studies of chronic duration, which provide a more appropriate data base for quantitative risk assessment. An ADI for fluridone of 0.092 mg/kg bw/day, or 6.44 mg/day for a 70 kg man was derived based on a NOEL of 9.2 mg/kg bw/day in a 2-year feeding study in rats and an uncertainty factor of 100 (10 for interspecies extrapolation and 10 to protect the most sensitive human within the population). A dosage of 25.2 mg/kg bw/day, which produced glomerulonephritis in this study, was the LOAEL.

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APPENDIX:

LITERATURE SEARCHED

This profile is based on data identified by computerized literature searches of:

CA SEARCH (Files 308, 309, 310, 311, 320)
TOXLINE
RTECS
OHM TADS
STORET
SRC Environmental Fate Data Bases
SANSS
AQUIRE

These searches were conducted in November, 1983. In addition, hand searches were made of Chemical Abstracts (Collective Indices 6 and 7th), and the following secondary sources were reviewed:

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